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Land use impacts on freshwater regulation, erosion regulation, and water purification: a spatial approach for a global scale level

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Abstract
Purpose Rarely considered in environmental assessment methods, potential land use impacts on a series of ecosystem services must be accounted for in widely used decision-making tools such as life cycle assessment (LCA). The main goal of this study is to provide an operational life cycle impact assessment characterization method that addresses land use impacts at a global scale by developing spatially differentiated characterization factors (CFs) and assessing the extent of their spatial variability using different regionalization levels.
Methods The proposed method follows the recommendations of previous work and falls within the framework and principles for land use impact assessment established by the United Nations Environment Programme/Society of Environmental Toxicology and Chemistry Life Cycle Initiative. Based on the spatial approach suggested by Saad et al. (Int J Life Cycle Assess 16: 198–211, 2011), the intended impact pathways that are modeled pertain to impacts on ecosystem services damage potential and focus on three major ecosystem services: (1) erosion regulation potential, (2) freshwater regulation potential, and (3) water purification potential. Spatially-differentiated CFs were calculated for each biogeographic region of all three regionalization scale (Holdridge life regions, Holdridge life zones, and terrestrial biomes) along with a nonspatial world average level. In addition, seven land use types were assessed considering both land occupation and land transformation interventions.
Results and discussion A comprehensive analysis of the results indicates that, when compared to all resolution schemes, the world generic averaged CF can deviate for various ecosystem types. In the case of groundwater recharge potential impacts, this range varied up to factors of 7, 4.7, and 3 when using the Holdridge life zones, the Holdridge regions, and the terrestrial biomes regionalization levels, respectively. This validates the importance of introducing a regionalized assessment and highlights how a finer scale increases the level of detail and consequently the discriminating power across several biogeographic regions, which could not have been captured using a coarser scale. In practice, the implementation of such regionalized CFs suggests that an LCA practitioner must identify the ecosystem in which land occupation or transformation activities occur in addition to the traditional inventory data required—namely, the land use activity and the inventory flow.
Conclusions The variability of CFs across all three regionalization levels provides an indication of the uncertainty linked to nonspatial CFs. Among other assumptions and value choices made throughout the study, the use of ecological borders over political boundaries was deemed more relevant to the interpretation of environmental issues related to specific functional ecosystem behaviors.

Keywords Characterization factors · Ecosystem quality · Ecosystem services · Global scale · Land use · Life cycle impact assessment (LCIA) · Regionalization · Spatial differentiation

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1 Introduction

Life cycle assessment (LCA) is a decision-making tool that is widely used in industries to assess the potential environmental impacts of a product throughout its entire life cycle. While it is well established that land use impacts are highly relevant from an environmental perspective, life cycle impact assessment (LCIA) methods are still in the early stages of development. The preliminary approaches advanced and adopted in several food study cases (Cederberg and Mattsson 2000; Berlin 2002; Thomassen et al. 2008; Basset-Mens et al. 2009; Roy et al. 2009) were mainly inventory data-related, simply reporting the amount of land occupied or transformed within a time period (Baumann et al. 1992; Heijungs 1992; Fava et al. 1993). More sophisticated methodologies were then developed, introducing the concept of indicators that include the effects of land quality changes over time (see Blonk et al. (1997), Lindeijer et al. (1998), and Lindeijer (2000)). However, modeling approaches tend to propose methods with site-generic characterization factors (CFs) that are limited to one geographical scope (mainly Europe) or restricted to a particular impact pathway, namely biodiversity or biomass production. Several methods and approaches have been recently suggested to address regionalization issues by developing explicit datasets (Mutel et al. 2011; Pfister et al. 2011); however, from an operational standpoint, the specific geographic information in life cycle inventories (LCI) cannot always be accounted for, making it impossible to differentiate land use-related impacts on a worldwide scale based on the location of use. To overcome this limitation, the need to guide and harmonize impact indicators is extensively justified in the literature (Müller-Wenk 1998; Udo de Haes et al. 2002; Milà i Canals et al. 2007a).

While land use has major impacts on biodiversity, it also constitutes a primary source of soil degradation (Tolba et al. 1992), which affects an ecosystem's structure and, consequently, its functional capacities, generating impacts on a broad range of ecosystem services through the degradation of the ecological quality of the soil (Foley et al. 2005). These impacts are intended to decrease the potential services that ecosystems can provide to sustain human wellbeing. Rarely considered within environmental assessment methods promoting sustainable development such as LCA, the role of resources and the potential impacts on ecosystem services must be accounted for (Maes et al. 2009; Zhang et al. 2010a, b). Their contributions to supporting the anthropogenic land use activities upon which human wellbeing is dependent have been considerably demonstrated in the literature for several years (de Groot 1992; Daily 1997; MEA 2005). It becomes crucial to properly integrate them and assess their environmental performances.

In view of this need and following the principles and recommendations for the calculation of land use impacts on biodiversity and ecosystem services on a global scale established by the United Nations Environment Programme (UNEP)/Society of Environmental Toxicology and Chemistry (SETAC) Life Cycle Initiative (Koellner et al. 2013), this paper presents an expanded approach of the method proposed by Saad et al. (2011). The initial paper aimed to develop a spatial characterization model to address land use impacts on a set of soil ecological functions within a Canadian application context. In an effort to comply with the principles and recommendations of the UNEP/SETAC Life Cycle Initiative Working group on land use LCIA (Koellner et al. 2013) for a global scale characterization, this study aims to provide an operational LCIA characterization method that addresses land use impacts on ecosystem services indicators (with a focus on soil ecological functions) by: (1) developing spatially differentiated CFs for a worldwide scale and (2) assessing the extent of their spatial variability using different regionalization levels.

2 Materials and methods

The proposed impact assessment method follows the recommendations of previously published work and falls within the framework and guidelines established by the UNEP/SETAC Life Cycle Initiative working group on land use LCIA (phase 1 (Milà i Canals et al. 2007a) and phase 2 (Koellner et al. 2012, 2013)).

A description of the general characterization modeling scheme and the choice of ecosystem quality indicators is provided (Section 2.1). The choice of several key elements needed to develop spatially differentiated CFs is then discussed (Section 2.2), including (1) the impact characterization model and calculation, (2) the choice of the baseline reference situation, (3) the selected regeneration time, and (4) the level of biogeographical differentiation and how data were collected and managed to cover a global scale level.

2.1 Characterization modeling scheme and indicators

The intended impact pathways that were modeled pertain to impacts on ecosystem services damage potential (ESDP). As proposed in the guideline paper (Koellner et al. 2013), the focus is on three major ecosystem services, which are reported by the Millennium Ecosystem Assessment (MEA 2005) to be significantly modified due to anthropogenic activities and, consequently, degraded: (1) erosion regulation potential (ERP), (2) freshwater regulation potential (FWRP), and (3) water purification potential (WPP). Many of these valuable ecosystem services are fulfilled by a number of soil ecological functions.

The same approach suggested by Saad et al. (2011) was applied in this study to provide a global application context. More specifically, the method proposed by Baitz (2002) that

was further developed into a calculation tool model, LANCA® (LANd use indicator value CAlculation) (Beck et al. 2010),[1] was used to assess the influence of different land use activities on soil ecological functions, which contribute to ecosystem services at a larger scale. Thus, the following four potential impact indicators were selected to address ecosystem services damage potential:

1. Groundwater recharge is a potential impact indicator for FWRP. It is measured in millimeters of water recharged annually (in millimeters per year) and represents the soil's ability to recharge groundwater in order to regulate peak flow through the magnitude of runoff and aquifer recharge.
2. Erosion resistance is a potential impact indicator for ERP. It is measured in tons of soil eroded per hectare per year (ton/(ha×year)) and represents the ability of a terrestrial ecosystem to withstand soil loss through erosion.
3. Physicochemical filtration is a potential impact indicator for WPP. It is measured in centimoles of cation fixed per kilogram of soil ($cmol_c/kg_{soil}$) and represents the soil's ability to act as a sorption matrix and to adsorb dissolved substances.
4. Mechanical filtration is another potential impact indicator for WPP. It is measured as the rate of water passing in a given time unit (in centimeter per day) and represents the soil's capacity to mechanically clarify a suspension through soil infiltration and supply a cleansing action to ensure groundwater protection.

Both, physiochemical filtration potential and mechanical filtration potential are seen as complementary indicators for WPP.

The LANCA® model requires that LCA practitioners collect and parameterize nine input parameters, reported in Table 1, that represent the condition of the environment to calculate the respective land use impact indicators. This approach is resource intensive (the LCA practitioner must collect all the input data as inventory flows) and can only be applied to foreground processes, when the information is available. This paper outlines a more practical solution to comply with the recommendations by Milà i Canals et al. (2007a) and Koellner et al. (2013) to link the impact assessment with LCI data with regards to the area, the type of occupation (or transformation) and the time of occupation (or the type of transformation). The method consists in developing a set of CFs (per land use type) specific to each biogeographic region (biome). Each set of CFs is calculated by calibrating the LANCA® model according to the average conditions of a given biogeographic region. The parameterization on a region level is supported by a geographic information system, ArcGIS 9.3 (ESRI 2012), interpolating spatially-resolved databases on soil properties, landscape, and climatic conditions. The LANCA® model computes averaged biome-specific input parameters into soil ecological function impact indicators. The difference between the baseline reference state and the outputs yields a set of CFs for each biome and land use type. A description of the LANCA® model and its algorithm shown as a step-by-step procedure is provided in Electronic supplementary material (ESM) and its complete documentation is given by Beck et al. (2010).

Figure 1 shows a simplified illustration of how the LANCA® model was used and paired in this study and how it fits into the global framework set out in the guidelines of the UNEP/SETAC Life Cycle Initiative for global land use impact assessment in LCA (Koellner et al. 2012; see Saad et al. (2011) for additional information).

2.2 Development of characterization factors

2.2.1 Impact characterization

Two types of land use interventions are considered in LCI: land occupation and land transformation. Their impacts are characterized based on the development of the so-called ecosystem quality (Q) curve, which is a key element that enables the assessment of potential changes in soil functional capacities over time in comparison to a chosen baseline reference state. In this paper, Q represent each of the impact indicators presented in the previous section.

Occupation impacts maintain land Q at a given level, postponing the regeneration phase by a period equal to the occupation time. Transformation impacts imply modifications in a soil's physical characteristics to meet the requirements of a new occupation process. Also referred to as irreversible impacts (Van der Voet 2001; Lindeijer et al. 2002; Milà i Canals et al. 2007b), permanent Q changes correspond to the net difference between the relaxed state and the initial, pretransformed one. However, considering a reasonable time span in which land use potential impacts are integrated, a modeling period of 500 years is defined following the recommendation of Koellner et al. (2013). This modeling choice considers a full ecosystem recovery within the modeling period ($Q_{relax} \approx Q_{initial}$). Therefore, permanent impacts are not expected to occur, and only land occupation and transformation impacts are accounted for within the scope of this study. Although allocation of transformation impacts among users as well as the linear regeneration assumption made can be challenging in land use modeling (see Koellner et al. (2013) and Pfister et al. (2010)), these key issues are outside the scope of this study.

[1] Online publication for the method description: http://publica.fraunhofer.de/eprints/urn:nbn:de:0011-n-1435418.pdf

Table 1 Spatially resolved input parameters for the generic world and the terrestrial biomes scale level

Input parameters		Data range		Source
		World average	Individual biomes	
Soil properties	Soil texture	Loam	All	Harmonized Soil Database data sets (30 arc-second raster database)[a]
	Organic matter content (%)	4.41	1.08 to 11.70	Harmonized Soil Database data sets (30 arc-second raster database)[a,b]
	Gravel content (%)	9.73	6.35 to 16.32	Harmonized Soil Database data sets (30 arc-second raster database) [a]
	Cation exchange capacity (CEC) ($cmol_c$/kg)	16.05	9.20 to 25.86	Harmonized Soil Database data sets (30 arc-second raster database)[a]
	pH	6.23	4.94 to 7.74	Harmonized Soil Database data sets (30 arc-second raster database) [a]
Landscape and climatic conditions	Depth to groundwater (m)	Value fixed at 3 m[c]		
	Annual precipitation rate (mm/year)	697.22	69.81 to 3,290.34	Terrestrial Ecoregions Base Global data sets (terrestrial ecoregions database)[d]
	Annual evapotranspiration rate (mm/year)	465.66	85.69 to 1,553.12	Terrestrial Ecoregions Base Global data sets (terrestrial ecoregions database)[d]
	Slope (°)	3.5	0 to 30	HYDRO1k Elevation Derivative Database (30 arc-second digital elevation model)[e]

[a] FAO et al. (2008)

[b] Organic matter content can range from 48 to 58 % (Nelson and Sommers 1996) and was calculated based on an approximate factor of 1.8 times soil organic content. The latter was calculated using the Harmonized Soil Database data sets

[c] Average value with high uncertainty due to data gaps (Stone and Myslik 2007)

[d] Olson et al. (2001)

[e] U.S. Geological Survey and Earth Resources Observation and Science (EROS)

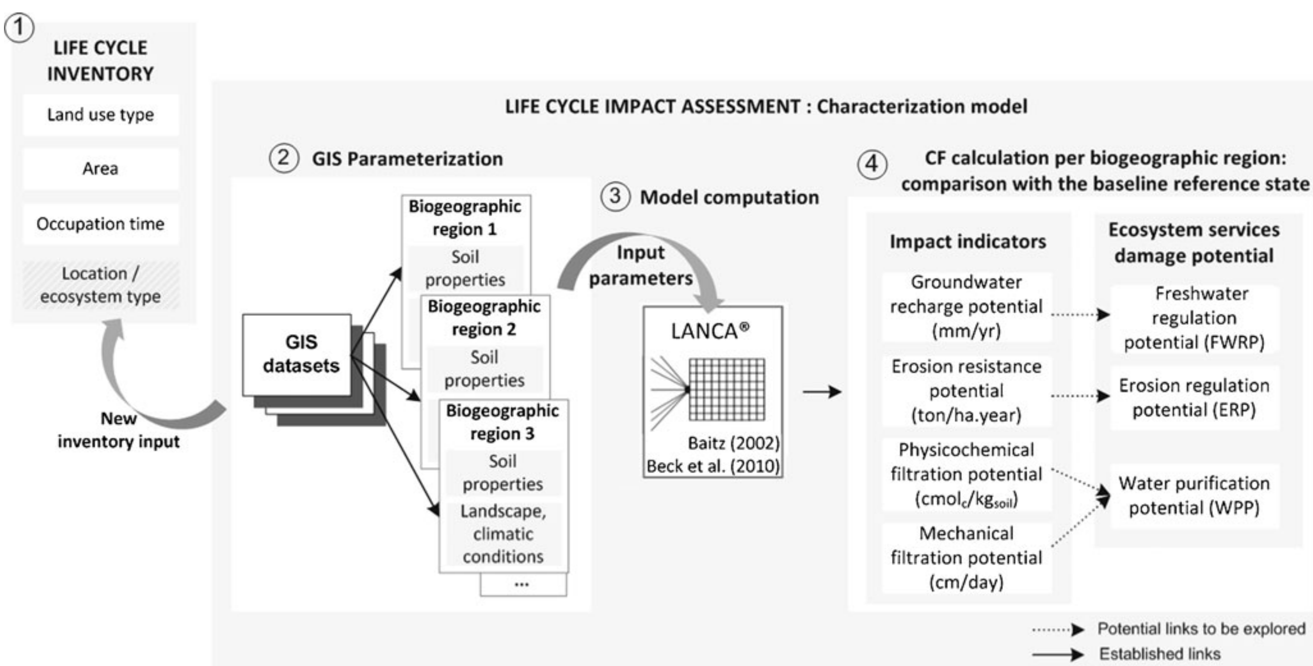

Fig. 1 Illustration of the characterization modeling scheme for land occupation and land transformation linking LCI inputs to ecosystem services potential damages

Based on the development of a Q curve over time (see ESM Fig. S2.1), impact scores for land occupation (I_{occ}) and land transformation (I_{trans}) are given in Eqs. (1) and (2), respectively:

$$I_{occ} = A \times t_{occ} \times CF_{occ}, \quad \text{where } CF_{occ} = Q_{ref} - Q_{LU} \qquad (1)$$

$$I_{trans} = A \times CF_{trans}, \quad \text{where } CF_{trans} = (Q_{ref} - Q_{LU}) \times 0.5 \times t_{reg} \qquad (2)$$

The inventory flow is $A \times t_{occ}$ for land occupation and A for land transformation; the parameter A (square meters) is the occupied or transformed area; t_{occ} (year) is the duration of the occupation stage (t_2 to t_3 from ESM Fig. S2.1) and t_{reg} (year) is the regeneration time needed for the ecosystem to reach a quasinatural state (t_3 to t_4 from ESM Fig. S2.1). With regards to each impact indicator, the CF measures the difference between the Q of the baseline reference (Q_{ref}) at the relaxation time stage and the land use type (Q_{LU}). This means that negative values of CF mean an increase in the respective ecosystem services and positive values express the reduction of a service being a negative ecological impact.

2.2.2 Baseline reference situation

Potential natural vegetation (PNV) was chosen as the reference state in this study. It describes the vegetation that would develop if all human influences on the site and its immediate surroundings would stop at once reaching a terminal stage (Westhoff and Van der Maarel 1973). It represents a more theoretical concept referred to as a prediction based on the most mature vegetation stage observed on a given site (Farris et al. 2010; Loidi et al. 2010). Due to the forces of nature, a potential land cover comparable to a quasinatural state would be reached after land abandonment through spontaneous regeneration. Hence, in the case of attributional LCA, choosing the PNV as a baseline for assessing land use impacts is consistent with the "non-use of the same land" default reference. However, the choice of the baseline reference is not arbitrary and, depending on the goal and scope of the study, the alternative land use system to which the study system is compared may also be suggested in a consequential modeling approach (Milà i Canals et al. 2007a).

2.2.3 Regeneration time

Regeneration time (t_{reg}) depends not only on the intensity of the land use type during the transformation phase (Müller-Wenk 1998) and the size of the area (Milà i Canals et al. 2007a) but also on the dynamics of the vegetation cover in specific biogeographic unit conditions. Regeneration times vary according to the type of land use intervention, impact pathway, and ecosystem type. Certain publications have listed regeneration time estimations (Müller-Wenk 1998; Koellner and Scholz 2007; Schmidt 2008). Within this study, data specific to each ecosystem service and its functioning were assumed based on estimates suggested by van Dobben et al. (1998) at a global scale. van Dobben et al. (1998) proposes rough estimates of regeneration times varying as a function of latitude and altitude for different regional ecosystems (additional information is provided in ESM). A more detailed discussion on regeneration time selection is given in Koellner et al. (2013).

2.2.4 Biogeographic differentiation and data collection

Highly influenced by the geographic conditions of the area in which the intervention takes place, the impact magnitude depends on many factors, namely climate conditions, soil type properties, slope, vegetation patterns, and use type (Milà i Canals et al. 2007a). The development of regionalized CFs considers a spatially-explicit approach at a global scale level. Two spatial resolution scheme models were created to address the global context.

First, the Holdridge life zones classification was used based on different biogeographic characteristics considering three dominant factors: annual precipitation, potential evapotranspiration ratio, and biotemperature (Holdridge 1947). This bioclimatic model refers to a diverse combination of climate conditions associated with a given vegetation cover type. Two different hierarchical scale levels were studied based on (1) the higher level of nine global Holdridge regions and (2) a more detailed classification level of 38 Holdridge life zones. The Holdridge system was deemed significant, since it plays a key role in discerning a distinct climatic geographic distribution and provides a simple and unambiguous organization of nature at a global scale (Ricklefs and Miller 2005).

Then, following the guidelines of the UNEP/SETAC Life Cycle Initiative for global land use impact assessment in LCA (Koellner et al. 2012), CF results were aggregated into a broader classification for terrestrial biomes (Olson et al. 2001), which corresponds to level 3 of the five different regionalization levels and defines a mapping system of 14 terrestrial biomes based on biogeographic regions delineating large units of land with a distinct assemblage of natural communities sharing similar environmental conditions and ecological dynamics. In addition, a generic nonspatial model that considers the world as one single huge biogeographic region (excluding Polar Regions where there is no vegetative soil cover) was also developed for comparative purposes. The purpose was to compare different levels of geographical differentiation with respect to the distribution of CFs.

For each biogeographic region, all input parameters required to calibrate the LANCA® model (Beck et al. 2010) are reported in Table 1. Data were collected from accessible databases and then managed using ArcGIS 9.3 (ESRI 2012).

Similarly to the spatially-resolved mean values that were calculated for each unit of all spatial models, average values weighted with regards to the surface areas were used for the nonspatial model. Serving as the reference state, a PNV type was identified for each biogeographic region area based on the output maps of the BIOME3 (Haxeltine and Prentice 1996) and BIOME4 (Kaplan et al. 2003) terrestrial biosphere models, which were designed to simulate the global vegetation distribution of major PNV types on a world scale level. The quality of those PNV were then determined using the LANCA® model (Beck et al. 2010).

3 Results and discussion

By combining the results of the LANCA® model for each Q_{LU} with each Q_{ref}, spatially-differentiated CFs were calculated according to Eqs. (1) and (2). This was done for each biogeographic region of all three regionalization scale levels (Holdridge life regions (n=9), Holdridge life zones (n=38), and terrestrial biomes (n=14)), along with nonspatial CFs representing the world average. CFs for land occupation and land transformation were calculated for all four impact indicators and for seven broad land use types being: (1) artificial green urban, (2) managed forest, (3) grassland, (4) pastures, (5) permanent and annual crops, (6) shrubland, and (7) urban. The land use typology that was used was structured to represent the effects induced by anthropogenic land uses with different land covers and sealed surface intensities that influence the ecosystem's functional capacity behavior. Detailed tables of CFs for spatial and nonspatial models and for both land occupation and land transformation are given in ESM. In general, CFs measure the difference in ecosystem quality of a specific land use type in comparison to a baseline reference state, where a positive value expresses a reduction in the corresponding soil ecological function potential and a negative value indicates a land use activity credit for the performance of the soil functional potential.

Spatial variability is only assessed for CF_{occ} specifically developed for each land use type. CF_{trans} can then be calculated per land use type proportionally to CF_{occ} by a factor of $0.5 \times t_{reg}$ as shown in Eq. (2). To facilitate the interpretation of the results and the analysis of the spatial variability between each regionalization scheme level, resulting CFs are reported in boxplots grouped by land use type and biogeographic unit, indicating their statistical distribution. Other measures of dispersion including a calculation of the average absolute deviation were also carried out. Land use impacts results on ecosystem services indicators are firstly discussed (Section 3.1). Spatial variability of characterization factors, their extent, and the influence from different regional scale levels are then assessed (Sections 3.2 and 3.3). A practical application of such characterization factors and their implementation in LCA is finally provided (Section 3.4).

3.1 Land use impacts on ecosystem quality

The regionalized assessment enables the comparison of impacts from the same type of activity among a range of biogeographic regions that differ by their soil properties, landscape, climatic conditions, and vulnerabilities as well as PNV. As an illustrative example, Fig. 2 shows the magnitude of groundwater recharge potential reduction for urban land use on a global scale. While additional maps can be found in ESM, similar maps can be further created for all four impact indicators and for each land use type assessed. Developing such

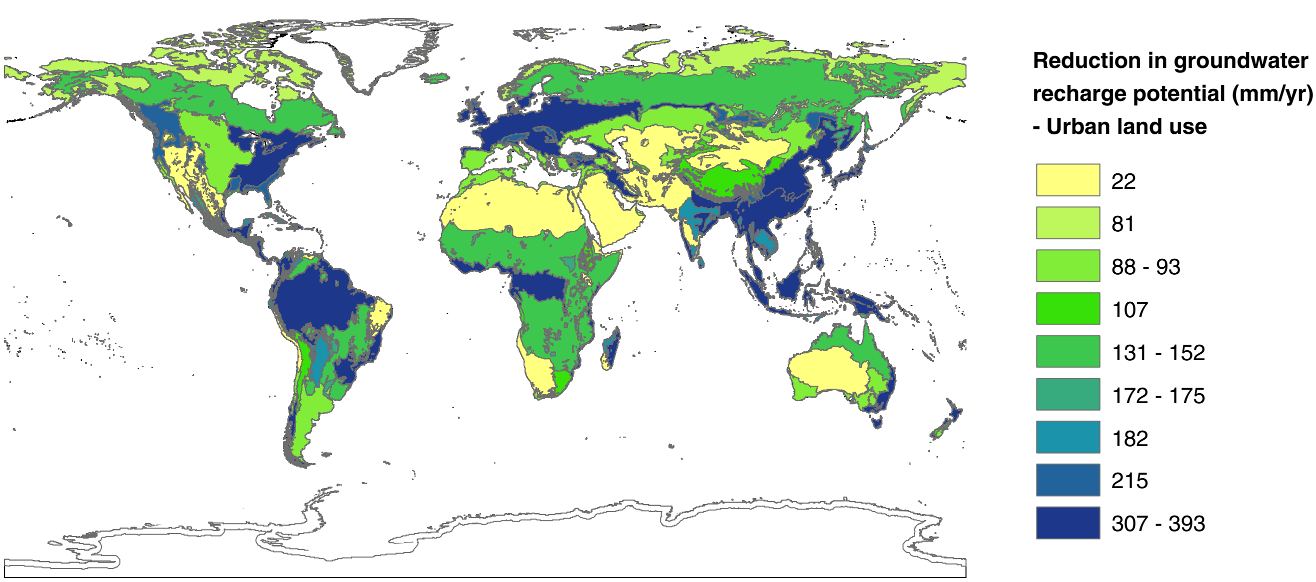

Fig. 2 CFs magnitudes as a reduction of groundwater recharge potential for urban land occupation using the terrestrial biomes regionalization level

maps tends to be useful since hotspots can be identified to highlight the ecosystems that are more vulnerable to land use activities than others depending on their biogeographic locations. However, it is important to bear in mind that the results are based on averaged data on soil properties landscape and climatic conditions over the entire biome.

Depending on the location of use, land occupation impacts can vary from 22 mm/year in the deserts and xeric shrublands biome to 393 mm/year in the (sub)-tropical moist broadleaf forests biome, as opposed to the world area weighted average CF (considering one single biogeographic region) of 155 mm/year. This variation of more than 1 order of magnitude demonstrates the need for a regionalized assessment when addressing impacts from a specific land use type, distinguishing between the world's various ecosystem types.

Considering that absolute ecosystem quality changes are calculated for the CFs presented in this study, the results must be interpreted with caution. Indeed, dry regions indicate a much lower impact figure that can be argued when compared to relative impact (normalized to the reference), since it considers that the regions have low annual rainfall and may consequently result in a higher impact figure. Land use is a driver for a change in ground and surface water quantity and quality and precipitation water stored as soil moisture. The modification of the hydrological balance following land transformation or occupation could be integrated in a life cycle impact assessment framework as it corresponds to a modification of the amount and quality of water that reaches the groundwater and surface water (equivalent to a use of the corresponding water). Example of water use impact assessment methods are provided by Kounina et al. (2013).

3.2 Assessment of the spatial variability of characterization factors

Figure 3 shows the CF results of changes in groundwater recharge potential (a positive value indicates a reduction and a negative value an improvement) in a range of land use types calculated using the terrestrial biomes scale level. Box plots illustrate the statistical distribution and display the full range of variation from minimum to maximum CFs, the common interquartile range (25th percentile and 75th percentile), the average absolute deviation and typical values of central tendency, and the median and the mean values. Similar graphs for the three remaining impact indicators are given in ESM.

A comparison of each set of median and mean values from Fig. 3 clearly showed a difference in impact magnitude among the series of seven land use types. Urban land occupation activity is shown to be the most impacting land use type since it reduces groundwater recharge potential by 165 mm/year as compared to the baseline reference situation. Established based on a common water balance cycle, annual groundwater recharge is highly dependent on hydrological components that consider the difference between precipitation, evapotranspiration, infiltration, and surface runoff. Under urban development, the water cycle is expected to be highly distorted. Annual infiltration and evapotranspiration rates decrease as the level of urbanization increases due to the effects of impervious areas. This increases surface runoff, limiting groundwater inflow and resulting in a reduction of annual groundwater availability for potential recharge (Erickson and Stefan 2009; Lee et al. 2010). This decreasing infiltration rate also explains how mechanical filtration potential is highly affected by an urban land use type, restraining the recharge ability and indicating a larger reduction among all other land use types (ESM Fig. S3.3).

Other land use types such as agricultural use show a smaller reduction in annual groundwater recharge potential when compared to the PNV state (an average of 25 mm/year). A limitation of the LANCA® model (Beck et al. 2010) is the estimate of the natural annual groundwater recharge brought about by different land use types, which considers only infiltrated precipitation, runoff, and evapotranspiration rates. Artificial recharge through irrigation is not accounted for, as this water withdrawal is already addressed by models assessing the potential impacts of water use. Over-irrigation

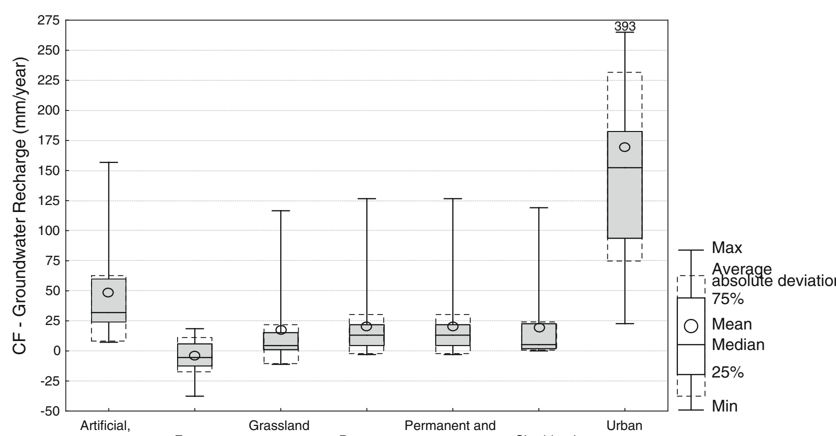

Fig. 3 Overall spatial variability of CFs for land occupation for Δgroundwater recharge potential using the terrestrial biomes regionalization level (reduction in groundwater recharge potential (CF>0), improvement in groundwater recharge potential (CF<0))

is not accounted for in consumptive water use because it is assumed to re-infiltrate and would therefore result in double counting (Bayart 2008; Pfister et al. 2009). While results from global simulation, field observations, and local measurements suggest that agricultural land typically increases groundwater recharge (Scanlon et al. 2005; Zhang and Schilling 2006; Rost et al. 2008), the underestimation of the annual recharge rate calculated by the LANCA® model (Beck et al. 2010) and its limitation may explain the discrepancies as it is enable to model this issue at a global level.

Erosion resistance potential is estimated as the difference in annual erosion rates between the PNV state and the land use activity and calculated with the Universal Soil Loss Equation (Wischmeier and Smith 1978). Mainly affected by vegetation cover, soil type, and topography, a smaller soil loss is observed for quasinatural dense cover such as grassland, resulting in a small reduction in erosion resistance potential. Conversely, agricultural land use types result in a higher reduction in erosion resistance potential since their erosion rate is greater than the baseline reference (ESM Fig S2.2). These trends in the results are consistent with the observations of Nelson et al. (2009). Although a distinction should be made between permanent crops and annual crops, the LANCA® model (Beck et al. 2010) is not able to differentiate between the different agricultural practices, which is an important limitation from the algorithm of this model.

In addition, high annual erosion loss rate observed for urban use, i.e., paved areas, are explained by a model artifact. The LANCA® model (Beck et al. 2010) assumes a maximum erosion rate to account for the permanent soil removal. However, this should be related to land transformation rather than occupation, potentially leading to double counting.

Moreover, it is important to note that the method proposed in this study is based on average parameters representing broad biogeographic regions, which may not be always comparable to results from local measurements and/or simulations performed on a site-specific area. A higher resolution might be reached in future work to better capture distinct local conditions and seasonal variability of rainfall as done in other impact categories such as water use (Pfister et al. 2009) and acidification (Roy et al. 2012).

Although ecological impacts can vary tremendously depending on the intensity of land use (Koellner and Scholz 2008), the LANCA® model does not yet enable a finer distinction between other levels of land use classes such as intensive and extensive activities.

Finally, an indication of the uncertainty for each land use is provided by the measure of dispersion, being the average absolute deviation (also known as average deviation). It shows the deviation of CF results from a central tendency, in this case the median value, obtained using a terrestrial biomes regionalization level (see ESM for a more detailed description). Therefore, when the information regarding the biogeographic region system location is not available at the inventory level, the assessment estimates the model uncertainty linked to the spatial variability.

3.3 Extent of spatial variability and influence of different regionalization levels

A comparison of the spatial variability of CFs was assessed for three different levels of regionalization, both Holdridge life zone-based scales delineation (9 regions and 38 zones) and the 14 terrestrial biomes along with a nonspatial model of the world area averaged. Figure 4 illustrates the CF results distribution patterns for each land use type, indicating the extent of their spatial variability within and across different biogeographic regions.

The general trends of results presented in Fig. 4 indicate that, when compared to different spatial models, the CFs developed by nonspatial models can deviate for a large number of ecosystem types (i.e., biogeographic regions) up to factors of 7, 4.7, and 3 for the Holdridge life zones, the Holdridge regions, and the terrestrial biomes regionalization scales, respectively. World average results for groundwater recharge and erosion resistance capacities could underestimate between 25 and 50 % of the spatially-differentiated CFs, whereas for physicochemical and mechanical filtration, up to 75 % of the CFs spatial distribution tends to be overestimated.

Similar trends are observed in overall spatial variability among all three regionalization levels, systematically indicating a larger spread in the distribution of the results for the Holdridge life zone-based model. A finer level of regionalization such as the 38-unit Holdridge life zones increases the level of detailed assessment and, consequently, the discriminating power across several biogeographic regions, which could not have been captured using a coarser scale level.

The largest reduction in groundwater recharge potential was in the tropical wet forest Holdridge life zone (1,072 mm/year) and was up to three times greater than the reduction obtained in the (sub)-tropical moist broadleaf forest terrestrial biome where no clear distinction is made between the humid regions (see Fig. 4a). The effects of urbanization on such humid regions, characterized by a high annual precipitation rate, disrupt the water balance by diminishing the soil's water storage capacity and therefore reducing groundwater recharge. Negative CF results were observed for anthropogenic forestry land use (e. g. managed forest), suggesting a beneficial activity in dry regions such as the tropical dry and very dry forests as compared to the baseline reference state ($Q_{\text{use=forest}} > Q_{\text{PNV=tundra}}$).

While erosion resistance potential shows very small variability in CF results across different biogeographic region scales for natural and seminatural land use types, important differences may be noted for artificial and agricultural land use activities. A comparison of all three resolution scales

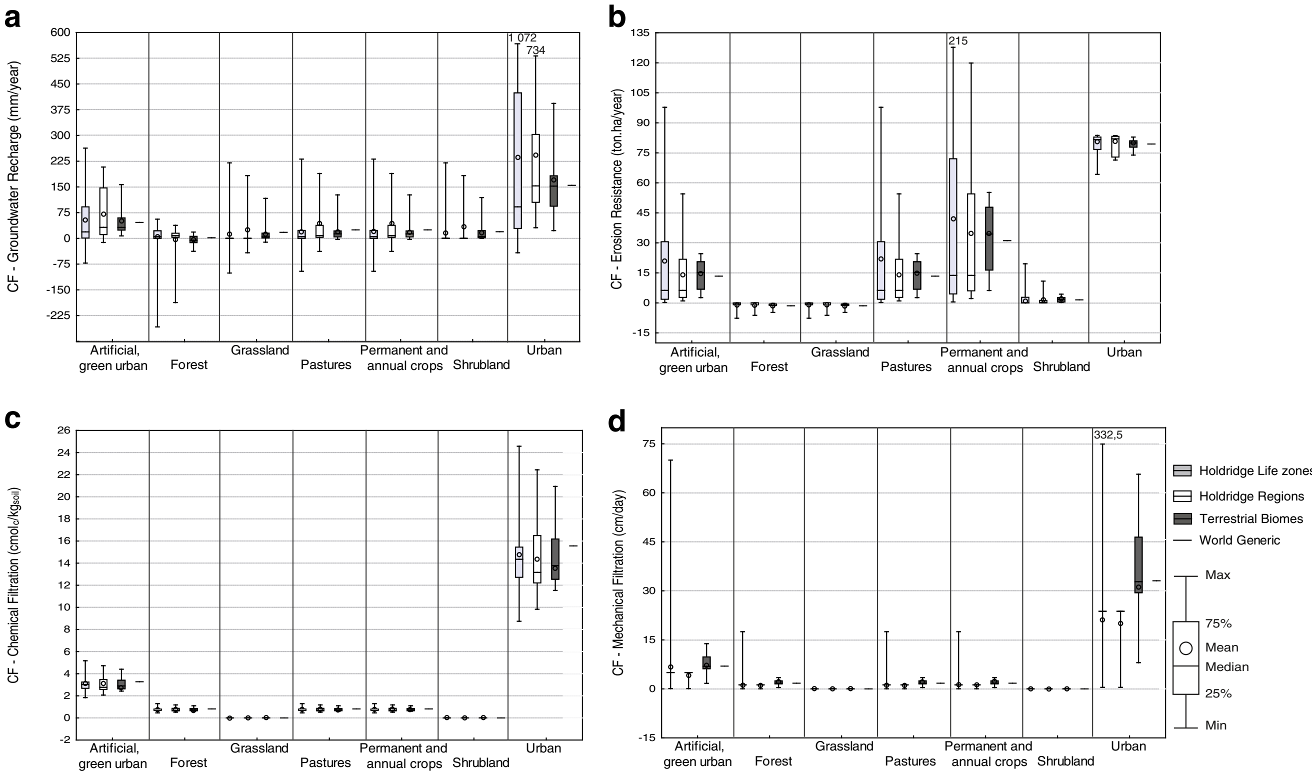

Fig. 4 Comparison of CFs estimates for land occupation from four regionalization level: the Holdridge life zones (*light gray box plot*), the Holdridge Regions (*white box plot*), the terrestrial biomes (*dark gray box plot*) and the World Generic nonspatial model (*plain* line) for four impact indicators: **a** Δgroundwater recharge potential, **b** Δerosion resistance capacity, **c** Δphysicochemical filtration, and **d** Δmechanical filtration (reduction is always indicated by CF>0, improvement is indicated by CF<0)

shows how moving from a coarser (terrestrial biome and Holdridge regions) to a finer level (Holdridge zones) induces a larger spread of up to 2 and 4.5 times in CF distribution across land use types (see Fig. 4b). The highest CF appears to be in the warm temperate Holdridge life zones, which are naturally characterized by a low erosion resistance capacity due to a weak- and fine-textured soil. Conversely, a dense ground vegetative cover helps improve erosion resistance by reducing potential soil loss. Steepness is also a key factor for erosion resistance and is best captured with a finer resolution level.

Differences in the overall variability of the filtration capacity reductions of spatial models are indicated for all land use types, except natural and seminatural ones, for which CFs are equal or close to zero. Thus, negligible reduction is associated with these activities when compared to the PNV baseline reference state. For the most impactful urban land use activity, the reduction in physicochemical filtration capacities ranged from 9 $cmol_c/kg_{soil}$ in tropical and desert regions with low CEC to 25 $cmol_c/kg_{soil}$ in boreal and taiga forests, where rich clayey textured soils are characterized by high CEC (Fig. 4c). Greater differences can be captured through mechanical filtration reduction results, which may have up to 16 times higher spatial variability in the distribution when referring to the finest spatial model, the Holdridge life zones (see Fig. 4d), since the estimated parameters for each soil filtration reduction potential indicator can be averaged out when using different scale levels, distorting the final values. Thus, inherent and site-dependant soil parameters, namely soil texture and slope, are more easily captured using a finer spatial model that accounts for many different ecosystem types.

3.4 Operationalization and application in LCA

In practice, accounting for spatial differentiation while developing CFs should be affordable, feasible in terms of data collection, and easy to implement in LCIA methods for which results interpretation is straightforward (Sedlbauer et al. 2007). When addressing site-dependant impact categories such as land use, and more particularly freshwater regulation, erosion regulation, and water purification, nine input parameters, including soil properties, landscape, and climatic conditions are required to calculate the CFs. In order to avoid tremendous data acquisition at the inventory level, CF development was calculated for archetypal situations based on the delineation of ecological units, each distinguishing different types of ecosystems and defined

by a distinct set of CFs. Consequently, an LCA practitioner must, in addition to the traditional inventory data required (e.g., land use activity and the inventory flow) identify the ecosystem in which land occupation or transformation activities occur. This issue of spatial variability and mapping from inventory to impact assessment spatial support has been deeply analyzed by Mutel et al. (2011).

On the one hand, CFs based on a spatial model that distinguishes between different broad biogeographic regions (i.e., terrestrial biomes or Holdridge life zones) is necessary, especially for large countries with several types of ecosystems. However, depending on the scope of the study, if the exact location is available or when land use activities and their related impacts become major contributors to the LCA study, a comprehensive assessment to support land management decisions should be conducted. To preserve the credibility of the results, site-specific CFs based on precise local parameters should therefore be calculated. Finally, when the location is unknown and the ecosystem type cannot be identified, global generic CFs (i.e., world area averaged model) can alternatively be used by acknowledging the associated uncertainty.

The CFs for the three resolution scale levels presented in this study consisted of estimates for broad biogeographic regions when accounting for land use impacts on a series of ecosystem services. However, one should bear in mind that such factors could underestimate the uncertainty related as the assessment only considers a limited number of regions (9 global Holdridge regions, 38 Holdridge life zones, and 14 terrestrial biomes). Indeed, the uncertainty decreases as the spatial resolution increases (i.e., a finer spatial scale).

4 Conclusions and recommendations

Differences in spatial variability are strongly linked to intrinsic and natural ecosystem characteristics that are highly dependent on the geographic conditions where the land use intervention occurs. When compared to the world average results, the variability of CFs among all three regionalization levels provides an indication of the uncertainty of nonspatial CFs as a factor of 0.5 to at least 1.5. Unlike the development of CFs for carbon sequestration potential (Müller-Wenk and Brandão 2010), a more detailed regionalization level is preferred when addressing land use impacts on the studied ecosystem services potential. However, though a finer resolution scale such as the Holdridge life zone-based model has proven to be more effective since it further discriminates the results, supplementary efforts are required by model developers to collect precise spatial-specific input parameters.

Several debatable assumptions and value choices are set out in this study. CF development is directly proportional to the Q difference. Moreover, the magnitude of the land transformation impacts is highly sensitive to regeneration time (Schmidt 2008). Therefore, CF_{trans} should only be regarded as a first estimate in an attempt to provide a comprehensive set of factors for various land use activities, land occupation, and land transformation. Nonetheless, additional and extensive research should focus on deriving regeneration time estimates for different impact pathways and ecosystem functioning potentials in particular. This will refine and improve CF_{trans} data.

Experts advocate the selection of ecological boundaries because of their relevance to the interpretation of ecosystem-specific environmental issues. While political boundaries are practical for simple and accessible data collection on a national scale, such boundaries are also bound by a random distribution on a spatial scale, often indicating discontinuous geospatial information and resource management jurisdiction that does not correlate with ecosystem function behaviors (Pintér et al. 2000).

Additionally, the choice of PNV as a baseline reference together with regeneration time estimates has been advanced in this study. Insofar as the baseline reference is not perceived as a benchmark for maximum ecosystem quality, the choice of PNV or seminatural state is appropriate for addressing land use impacts in LCA.

Only the spatial variability of CF distribution at a global scale was assessed in this paper. In order to better understand the input parameters and their influence on the final results, a more detailed analysis of variance (multiway MANOVA) using the LANCA® model was performed within the scope of the study published by Saad et al. (2011).

The choice of several impact indicators may limit the approach. This is generally not recommended in LCA since the need for aggregation or weighting becomes key to decision making. However, considering the nature of the ESDP modeled impact pathway and with respect to its multifunctional aspect, a comprehensive assessment framework has been set out to address a range of intended functional capacities based on appropriate impact indicators. It can also be further discussed for selecting and limiting the most relevant indicators. As highlighted in Saad et al. (2011), using the LANCA® model (Beck et al. 2010) is a first step in developing and integrating spatially-differentiated CFs for modeling land use impacts within LCA. Further research should focus on improving this model into a more sophisticated tool by overcoming its limitations. Other models that tend to be more specific in modeling particular a soil ecological quality change and ecosystem service due to land use could be used and tested such as Erosion Productivity Impact Calculation (Williams 1990), Chemicals, Runoff, and Erosion from Agricultural Management Systems (Knisel 1980), Water Erosion Prediction Project (Laflen et al. 1997), and Integrated Valuation of Environmental Services and Tradeoffs (Nelson et al. 2009; Tallis and Polasky 2009).

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